

**UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 5**

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SUBJECT: ACS Superfund Site Focused Ecological Risk Assessment (FERA), Griffith, Indiana, prepared by Jim Chapman, Ph.D., Ecologist, USEPA

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This memo summarizes the following aspects of the ACS Superfund Site FERA prepared by Jim Chapman:

- Conceptual site model
- Assessment endpoints
- Toxicity of PCBs
- Exposure models
- Recommended cleanup levels



Conceptual Site Model

Wetlands

The scope of this FERA has been defined by a previous ecological risk assessment of the ACS site (Weston, 1992) in which potential risks to on-site wetlands were identified. All subsequent work has focused on these wetland areas. The wetland areas are located west and northwest of the ACS facility. They consist of marsh communities overlapping somewhat with wet meadow and scrub-shrub communities. The marsh communities cover a majority of the wetlands and consist primarily of cattails. Wet meadow communities are located in small pockets, primarily in the north and east portions of the site. The wet meadows are distinguishable by low growing sedges and grasses. The scrub-shrub communities are located in wetland transitional areas, primarily along the north and west side of the ACS facility, between the facility and the main wetland areas (Montgomery Watson, 1997a).

Surface water in the wetland is highly variable depending on the time of the growing season and the rainfall. Typically, in the spring and fall, prolonged periods of inundation or saturation occur. Water levels may approach a depth of three feet towards the center of the wetland during these seasons. During the dryer summer and winter months, the water table in the wetland fluctuates as the area is inundated for several days to several weeks depending on the frequency of duration of rainfall events in the area (Montgomery Watson, 1997a).

Facility Inputs to Wetlands

Several drainage ditches on the site feed into the wetland areas. A small drainage channel extends from the southwestern portion of the ACS facility into the wetland to the west. Aerial photographs show that prior to 1970, this channel extended from inside the facility, west across the wetland, to the drainage ditch running north to south through the center of the wetland. Another channelized drainage ditch exists along the north and west side of the site. Historical aerial photographs indicate that the ditch was also channelized south through the center of the large

wetland. In addition, there is evidence that there had been direct runoff from the ACS facility to the north. All of these conduits for water could have served as inputs of contaminants into the wetland areas. Surface water, groundwater seep, soil, and sediment samples taken at the site indicated varying composition and concentrations of VOCs, SVOCs, particularly PCBs, and metals (Montgomery Watson, 1997a).

Contaminants and Media of Concern

This FERA will focus on the risk associated with exposure to sediments contaminated with PCBs. High PCB concentrations (13.1 to 125 mg/kg) have been detected in several sediment sampling locations and appear to be associated with an old surface water runoff route from the ACS facility (Montgomery Watson, 1997a). Elevated concentrations of PCBs have also been found in soil within localized areas of the site. Due to their low solubility in water, PCBs have not been detected in the surface water or groundwater seeps.

Several other classes of contaminants are present at the site, but these will not be the focus of this FERA. VOCs, SVOCs, and metals have been detected in the sediments and soils. However they are distributed fairly uniformly in the wetland and are not addressed as a primary concern. VOCs have been detected in groundwater and seep discharge. The seep is not considered to represent a long term impact because a groundwater interceptor trench will be constructed in the vicinity of the seep (Montgomery Watson, 1997a). Impacted groundwater itself is not emphasized due to its relative unavailability to ecological receptors. Elevated iron levels (323 to 3060 µg/L) have been detected in filtered surface water samples, but concentrations in this range are common in some wetland environments (Montgomery Watson, 1997a). Consequently, iron is not considered a contaminant of concern.

Exposure Pathways

Exposure to wetland sediment PCBs could occur through the following pathways:

- 1) sediment/soil → receptor (incidental ingestion)
- 2) sediment/soil → insect larvae/other invertebrates → receptor
- 3) sediment/soil → insect larvae → adult insects (adult insects may be aquatic, terrestrial or aerial) → receptor
- 4) sediment → benthic invertebrates/detritus → fish/crayfish → receptor
- 5) sediment/soil → multiple pathways → amphibians → receptor
- 6) sediment → water column → aquatic plants → receptor
- 7) sediment/soil → rooted aquatic/terrestrial plants → receptor

Pathways 2 and 3 are likely to result in the greatest exposures. Pathways 6 and 7 are probably insignificant (with the caveat that algal uptake may be significant). Pathway 4 would be a major exposure route in a purely aquatic system, but is of uncertain significance in a wetland that has standing water only part of the year. Pathway 5 is difficult to estimate without biosampling, and is not likely to indicate risk levels appreciably greater than those associated with pathways 2 and 3 (since much of the amphibian exposure occurs through insectivory). Risks to amphibians may be as or more significant than their role as an exposure pathway for other receptors, but the data base for estimating effects on amphibians is meager.

Assessment Endpoints

Rails have been chosen as the model ecological receptor. The rationale for focusing on rails (Family Rallidae) is several-fold. The wetland PCB contamination at the ACS site is concentrated in a relatively small area, therefore a receptor with a small home range is preferable for assessing potential ecological effects; rails are common inhabitants of marshes and have small home ranges; several rail species potentially utilize the ACS wetland; and most of the rail species have predominantly insectivorous diets, the expected primary exposure pathway for wetland PCBs.

The following species may occur in the ACS wetlands:

<u>Species</u>	<u>Primary diet</u> (Sanderson 1977; Martin, et al. 1951)
Virginia rail (<i>Rallus limicola</i>)	larval and adult insects, snails, crustaceans, and small fish; plant food typically comprises less than 10% of the diet during the reproductive season
King rail (<i>Rallus elegans</i>)	crustaceans (crayfish) and other aquatic animals including amphibians and small fish
Sora (<i>Porzana carolina</i>)	mollusks, insects, and seeds (plant food may predominate in freshwater marshes)
Black rail (<i>Laterallus jamaicensis</i>)	not well known, insects and other invertebrates

The Virginia rail is the recommended measurement endpoint because it feeds by probing for food (as opposed to gleaning from the surface) and much of its prey are themselves predaceous (Sanderson 1977). Thus, the target organism has significant contact with sediments, and it is vulnerable to the bioaccumulative and biomagnifying properties of PCBs as the contaminants move up the food chain.

According to the Technical Memorandum Phase II Wetland Investigation dated February 14, 1997 (Montgomery Watson, 1997b), mink were originally considered by the USEPA as the receptor of concern for the site based on the potential for biomagnification to occur through its food chain within the wetland. However, mink have a large home range, and no mink have been observed at or near the site. Consequently, the Virginia rail is a more appropriate assessment endpoint for the ACS site.

Toxicity of PCBs

Recent reviews of the ecotoxicity of PCBs include Bosveld and Van den Berg (1994), Barron, et al. (1995), Eisler and Belisle (1996), and Hoffman, et al. (1996). Effects on birds are emphasized in this summary consistent with the selected assessment and measurement endpoints.

PCBs have been associated with a range of adverse effects in wildlife including growth, neurobehavioral, hormonal, reproductive, embryotoxic, immunotoxic, and lethal effects. Certain PCBs have been shown to be mutagenic and carcinogenic in laboratory studies, but cancers in wildlife have not been correlated with environmental PCB exposures. Many, but not all, adverse effects appear to be mediated through the same mode of action as for dioxins,

and are therefore attributed to the dioxin-like coplanar congeners. However, non-dioxin-like congeners also may be responsible for toxic effects through different modes of action (Fisher, et al. 1998; Johansson, et al. 1998).

One of the most sensitive adverse effects in birds related to PCB exposure is reproductive. Reduced reproductive success results from increased embryo mortality (reduced hatchability), deformities, and chick mortality; delayed hatching; and reduced growth rates. These effects may occur at PCB doses less than the levels causing overt parental toxicity, however, sublethal neurobehavioral effects (parental inattentiveness) has been shown to contribute to the reduced reproductive success in addition to the direct effects on embryos and chicks. Common external deformities include beak, leg, toe and neck abnormalities. Internal effects include increased liver weight and abnormalities in thyroid, bursa of Fabricius (an organ in birds that functions similar to the thymus), and pituitary weights. Growth rates of chicks may also be depressed. Although PCBs may affect eggshell thickness at very high doses, this effect usually does not play a role in impaired reproductive success because the embryo and chick adverse effects occur at much lower doses. Edema (excessive accumulation of fluids) in embryos results in embryo or chick mortality, but there are questions whether this effect is caused by PCBs or by other environmental contaminants.

PCBs have also been associated with impaired immune functions, endocrine (hormonal) disruptions, and altered vitamin A regulation. PCBs have been shown to promote of hepatic (liver) cancers in rodents.

There are significant differences in PCB sensitivities between species. Of the bird species tested, chickens are the most sensitive, followed by pheasants/turkey, ducks, and gulls, in descending order.

Exposure Models

Three models were developed in order to provide several approaches to approximating rail exposure to wetland PCBs. These models utilize one of two PCB toxicity reference values (TRVs) (one derived from pheasant studies, the other from chicken studies) and one of two assumed rail diets (earthworms or nonearthworm invertebrates).

Diet

Virginia rails primarily eat animal prey. As stated previously, their diet consists mainly of larval and adult insects, snails, crustaceans, and small fish (Sanderson 1977; Martin, et al. 1951). Both of the assumed diets in the models are consistent with the described diet of the rails. Although earthworms are not specifically mentioned in the literature as common food sources, they serve well as a reasonable approximation of rail prey. The earthworm diet could be deemed a more protective (or conservative) estimation of PCB exposure than the nonearthworm invertebrate diet as earthworms often exhibit greater bioaccumulation of contaminants than do other invertebrates. In all models, the stated diet was assumed to comprise all food intake by the rails (i.e. in the models, 100% of the birds' diet consists of the stated prey).

TRVs

The pheasant TRV is based on research done by Dahlgren et al. Dahlgren et al. (1972) assessed the effects of orally administered Aroclor 1254 on reproduction in the ring-necked pheasant. Pheasants were individually dosed once per week, for 16 weeks, via gelatin capsule at rates of 0, 12.5, and 50 mg/week for females and 0 and 25 mg/week for males. Egg production, egg fertility, egg hatchability, survivability, and growth of chicks through 6 weeks post-

hatch were monitored. Significant reductions in hatchability were reported among eggs from the females treated with 12.5 or 50 mg Aroclor 1254 per week. Egg production and chick survivability were significantly reduced among hens administered 50 mg Aroclor 1254 per week, but not among hens administered 12.5 mg per week. No effect of Aroclor 1254 administration on egg fertility or on chick growth was observed. Using a female ring-necked pheasant body weight of 1 kg (Nelson and Martin, 1953 in USEPA, 1995), a value of 1.8 mg/kg-day (12.5 mg/week) can be inferred from this study for the NOAEL for egg production and chick survivability as well as for the LOAEL for egg hatchability. A LOAEL was determined by dividing the NOAEL value by 10 yielding a LOAEL of 0.18 mg/kg-day.

The chicken TRV is based on a study of chicken (*Gallus domesticus*) fed naturally contaminated common carp (*Cyprinus carpio*) collected from the Saginaw River, Lake Huron, MI (Summer, et al. 1996a, b). The carp were analyzed for total PCBs on the basis of the sum of Aroclors 1242, 1248, 1254 and 1260, which should more closely approximate a congeners-based total PCBs than would any single Aroclor analysis. Different treatment doses were obtained by diluting the carp with chicken feed. Summer, et al. (1996a) reported mean bodyweight and daily PCB consumption ($\mu\text{g}/\text{hen}$) for biweekly intervals by treatment. For the purposes of this risk assessment, overall mean bodyweights and daily PCB consumption were calculated for the interval of weeks 1 through 8 following the onset of dietary exposure to contaminated carp (the duration of the experiment excluding the 2-week acclimation period), and the resulting values were used to calculate bodyweight-normalized PCB ingestion rates for each of the treatments. The results were checked by calculating PCB ingestion rates through a second procedure: the reported dietary PCB concentrations (single value for each treatment) were multiplied by the mean food ingestion rates for weeks 1 through 8 post-exposure. The two approaches were in close agreement. The treatment doses by the first procedure are 0.0159, 0.0415, and 0.361 mg PCBs/kg_{bw}-d for control, low-, and high-doses, respectively.

The TRVs were selected on the basis of reproductive effects reported in Summer, et al. (1996b). Hatchability decreased by 18 % in the high-dose treatment relative to the control (weeks 4 - 8 post-exposure), and total embryo/chick deformities increased 2.3 times (over the entire experimental period including the 2-week acclimation). Deformities increased 1.4 times in the low-dose treatment relative to the control, but hatchability was unaffected. The overall deformity rates were 17, 24, and 40 % for the control, low-, and high-doses, respectively. For the purposes of the present risk assessment, the high-dose treatment was selected as the lowest observed adverse effect level (LOAEL), that is, the lowest dose in which a toxic effect was detected. This was based on the decrease in hatchability and the large increase in deformities. The low-dose treatment was selected as the no observed adverse effect level (NOAEL), the highest dose in which toxic effects were not detected. This was based on the lack of effect on hatchability and the comparatively low increase in deformities, which was not considered to be biologically significant. In contrast, the more than doubling of deformity rates accompanied by decreased hatchability in the high dose treatment was considered a biologically significant effect.

The main uncertainty with this study is that the carp absorbed their contaminant loads in nature (Saginaw Bay), so they may have accumulated other contaminants in addition to PCBs. This means that the observed adverse effects may not be solely due to PCB exposure. For example, PCB congeners 77 and 126 accounted for 87 % of the TEQ of carp samples collected from the Saginaw River in 1983, but 2,3,7,8-TCDD accounted for an additional 12 % (Smith, et al. 1990). Conversely, because carp absorbed PCBs in nature, the congener profile should accurately reflect the changes that occur when PCBs are passed through a food chain (environment \rightarrow prey \rightarrow predator).

Models and Results

All three models follow similar formats. First, the following parameters are considered for all models:

- PCB sediment concentrations (mean = 17.33 mg/kg, upper confidence limit (UCL) = 31.12 mg/kg)
- Rail body weight (0.075 kg; value is based on females)
- Food ingestion rate (7.70 g/d; derived by using $0.31 \cdot BW(g)^{0.751}$ from USEPA, 1993)
- Normalized food ingestion rate (0.10 kg/kgBW-d; converted g to kg and expressed relative to BW)
- Fraction of sediment ingested (0.104; value for the American woodcock from Beyer et al., 1994)
- Area use factor (1; assumed all food is of site origin based on 0.5 acre home range from Berger, 1951)

For earthworm diet values, the following parameters are considered:

- Worm PCB bioaccumulation factor (WBAF) (11; a multiplier for earthworms from Kreis, et al., 1987)
- Worm dietary fraction (1; assumed earthworms comprise 100% of diet because animal food is most important; from Zimmerman, 1977)
- PCB earthworm concentrations (mean = 190.63 mg/kg, UCL = 342.32 mg/kg; derived by multiplying the worm PCB bioaccumulation factor by the mean or UCL PCB sediment concentration, respectively)

For nonearthworm invertebrate diet values, the following parameters are considered:

- Ratio of PCB concentrations in nonearthworm invertebrates and earthworms (RIWC) (0.125; derived from dioxin studies done by Martin et al., 1987 and Thiel, et al., 1987)
- Nonearthworm invertebrate PCB bioaccumulation factor (1.375; derived by $WBAF \cdot RIWC$)
- Nonearthworm invertebrate dietary fraction (1; assumed nonearthworm invertebrates comprise 100% of diet because animal food is most important; from Zimmerman, 1977)
- PCB nonearthworm invertebrate concentrations (mean = 23.83 mg/kg, UCL = 42.79 mg/kg; derived by multiplying the nonearthworm invertebrate bioaccumulation factor by the mean or UCL PCB sediment concentration, respectively)

Total mean and UCL PCB doses are then calculated in each model by adding ingestion of PCBs through food and sediment. Ingestion is calculated by multiplying (normalized food ingestion rate)*(mean PCB concentration in food source)*(food source dietary fraction)*(area use factor).

These doses are then compared to the TRVs based on both LOAELs and NOAELs. A hazard quotient for both mean and UCL doses is calculated by dividing the total dose by the LOAEL or NOAEL, respectively. A hazard quotient that exceeds 1 indicates that there is a potential risk to the target organism.

If a hazard quotient exceeds 1, separate protective remedial goals are based on the LOAEL and NOAEL and backcalculated by solving for a protective mean PCB sediment concentration using the following formula:

$$\text{LOAEL or NOAEL} / ((\text{normalized food ingestion rate}) \cdot (\text{area use factor}) \cdot (((\text{food source PCB bioaccumulation factor}) \cdot (\text{food source dietary fraction})) + (\text{fraction of sediment ingested})))$$

Model 1 assesses risk based on an earthworm diet and a pheasant TRV. Model 2 assesses risk based on a nonearthworm invertebrate diet and a pheasant TRV. Model 3 assesses risk based on a nonearthworm invertebrate diet and a chicken TRV. Table 1 summarizes the results of each model. Refer to Appendix 1 for more information regarding each of these models.

Table 1 - Model Results Summary

Model	Mean Hazard Quotient (LOAEL)	Mean Hazard Quotient (NOAEL)	UCL Hazard Quotient (LOAEL)	UCL Hazard Quotient (NOAEL)	Protective Remedial Goal (LOAEL)	Protective Remedial Goal (NOAEL)
1	10.98	109.82	19.72	197.21	1.58 mg/kg	0.16 mg/kg
2	1.46	14.63	2.63	26.27	11.85 mg/kg	1.18 mg/kg
3	7.31	62.69	13.13	112.57	2.37 mg/kg	0.28 mg/kg

All three models generate protective remedial goals (PRGs) that are relatively similar. Model 2 (nonearthworm invertebrate diet with pheasant TRV) yields slightly higher (less conservative) PRGs, but the PRG (NOAEL) is similar to model 1 and 3's PRG (LOAEL)s.

Recommended Cleanup Levels (PRGs)

Based on the relative concurrence of all three models, we propose that a PRG of 1 mg/kg should be used for the ACS site.

Jim may be contacted at 6-7195; John at 6-7180, if you have questions or comments. Please fill out the attached evaluation form and return it to Larry Schmitt, SR-6J. The information is used to assess and improve our services.

cc: Larry Schmitt, Section Chief, RRS #1

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